TEMPORAL AND SPATIAL VARIABILITY IN WATER QUALITY OF WETLANDS IN THE MINNEAPOLIS/ST. PAUL, MN METROPOLITAN AREA: IMPLICATIONS FOR MONITORING STRATEGIES AND DESIGNS

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Abstract. Temporal and spatial variability in wetland water-quality variables were examined for twenty-one wetlands in the Minneapolis/St. Paul metropolitan area and eighteen wetlands in adjacent Wright County. Wetland water quality was significantly affected by contact with the sediment (surface water vs. groundwater), season, degree of hydrologic isolation, wetland class, and predominant land-use in the surrounding watershed ($p < 0.05$). Between years, only nitrate and particulate nitrogen concentrations varied significantly in Wright County wetland surface waters. For eight water-quality variables, the power of a paired before-and-after comparison design was greater than the power of a completely randomized design. The reverse was true for four other water-quality variables. The power of statistical tests for different classes of water-quality variables could be ranked according to the predominant factors influencing these: climate factors > edaphic factors > detritivory > land-use factors > biotic-redox or other multiple factors.

For two wetlands sampled intensively, soluble reactive phosphate and total dissolved phosphorus were the most spatially variable (c.v. = 76–249%), while temperature, color, dissolved organic carbon, and DO were least variable (c.v. = 6–43%). Geostatistical analyses demonstrated that the average distance across which water-quality variables were spatially correlated (variogram range) was 61–112% of the mean radius of each wetland. Within the shallower of the two wetlands, nitrogen speciation was explained as a function of dissolved oxygen, while deeper marsh water-quality variables were explained as a function of water depth or distance from the wetland edge. Compositing water-quality samples produced unbiased estimates of individual sample means for all water quality variables examined except for ammonium.

1. Introduction

Effective monitoring and experimental designs for wetlands research should be based on a knowledge of factors contributing to the temporal and spatial variability of wetland water quality. Three issues must be addressed: (1) the choice of an

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efficient experimental design; (2) the choice of blocking factors or covariates to decrease the magnitude of background variability; and (3) the development of effective sampling strategies that account for within-wetland variability.

Two alternative experimental design can be used to examine treatment effects on wetland water quality: (1) a completely randomized design contained within one year to avoid effects of interannual variation, and (2) a before-and-after design (paired comparisons) to account for background variability in wetland condition. The relative efficiency of a paired comparisons (randomized blocks) design depends on the degree of geographic variability relative to temporal variability, or 'temporal coherence' of water quality variables among wetlands. Temporal coherence is 'the degree to which different locations within a region behave similarly through time' (Magnuson *et al.*, 1990). The two designs are equally powerful when interannual variability for paired comparisons is equal to variability among wetlands within a year (Sokal and Rohlf, 1981). Results of a test of the effects of two mosquito control agents, methoprene and *Bacillus thuringiensis* (BTI), on cattail marshes are used here to demonstrate the power of these two designs for different water quality variables.

Knowledge of continuous or categorical variables that explain a significant degree of variability among wetland sites can be used to create a more efficient sampling design. Background sampling error can be partitioned through use of blocking variables in a randomized block design or use of covariates in analysis of covariance (Gilbert, 1987; Keith, 1988). In the pre-treatment phase of a study of the effects of physical and chemical perturbations, factors contributing to spatial and temporal variability of wetland water quality were identified. Influence of both intrinsic wetland characteristics such as wetland type, hydrologic regime, water depth, and extrinsic watershed characteristics such as predominant land use were examined.

Strategies for sampling *within* wetlands must be designed to account for spatial heterogeneity of water quality. Several factors can interact to produce patterns of spatial variability in wetland water quality. Circulation of surface water within a wetland often is impeded by vegetation; a given wetland basin contains 'dead' zones with little or no circulation (Kadlec, 1988). In general, strong concentration gradients occur across wetland/upland or wetland/open water ecotones (Holland *et al.*, 1990). Although discharge/recharge patterns can change seasonally, groundwater discharge tends to occur more often around wetland/upland borders, with groundwater recharge occurring below bodies of open water (Carter and Novitzki, 1988; Siegel, 1988). Water depth and frequency of inundation strongly influence pH and redox potential and thus rates of decomposition, P solubility, and N transformations (Reddy and Patrick, 1975; Hossner and Baker, 1988; Reddy and Graetz, 1988; Gehrels and Mulamoottil, 1989). Subtle differences in elevation can reflect major differences in the frequency of sediment inundation, and these may correspond to discrete shifts in vegetation composition and in C/N dynamics within groundwater (Pinay *et al.*, 1980). Vegetative structure can affect nutrient cycling strictly by limiting light availability for photo-oxidation and algal growth. In addition, processes affecting nutrient cycling may be affected differentially by different plant species. Some wetland species transport oxygen through root parenchyma and create oxidized microzones within otherwise anaerobic sediments (de la Cruz *et al.*, 1989; Ernst, 1990). Vegetation of different species decays at different rates (reviewed by Brinson *et al.*, 1981) and the quality of available carbon can limit N transformations within sediments (Bowden, 1987).

Subsampling within wetlands can help to account for spatial heterogeneity, particularly if covariates or blocking factors are used to partition variability. However, the scale of subsampling must be chosen carefully to avoid biases due to spatial autocorrelation and to avoid collection of redundant information (Millard *et al.*, 1985; Legendre and Trousellier, 1988). Spatial autocorrelation is defined as the tendency of samples collected close to one another to be similar; spatial autocorrelation can invalidate the assumption of independent observations on which parametric statistics are based.

To examine the scale od spatial autocorrelation of water-quality variables within individual wetlands, a subset of rural wetlands was intensively subsampled, and data were subjected to geostatistical analysis. Hilton *et al.* (1989) found no significant differences in water-quality estimates among four sampling techniques for deep lakes, and only the vertically integrated composite technique produced a significant bias in water-quality estimates for shallow lakes. To determine whether compositing samples prior to analysis could yield a more cost-effective and unbiased estimate of average wetland water quality, water samples were collected along transects in eight of these wetlands. Samples were analyzed both individually and as composite samples.

2. Study Area

2.1. Twin Cities Metropolitan Area, Minnesota

The seven-county Twin Cities Metropolitan Area (TCMA) of east-central Minnesota (U.S.A.) encompasses a 7800 km^2 area (Figure 1). Land use is 27% urban, 43% agricultural, and 30% open space (Ayers *et al.*, 1985). The region is characterized by terminal moraines and glacial outwash with wetlands in areas of high water tables, in glacial kettle depressions, and along major rivers and associated tributaries (Ayers *et al.*, 1985).

Climate is continental, with mild, humid summers and relatively long, severe winters. Most rain comes in frontal storms or warm-weather convective storms, with May and June typically the wettest months and February the driest (Brown, 1984). Normal annual precipitation is 68.6 cm, including the water content of 111.8 cm average winter snowfall. Annual precipitation varied greatly during the study. In the drought year of 1989 annual precipitation was only 59.7 cm, while

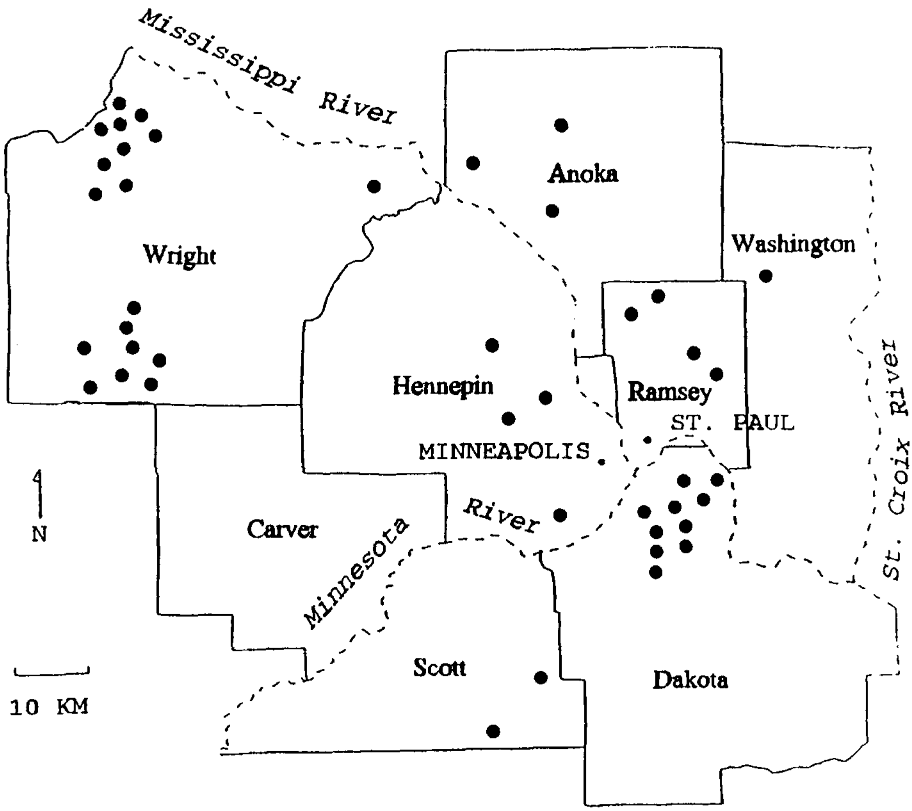

Fig. 1. Location of study site wetlands in the 8-county Minneapolis/St. Paul metropolitan area and adjacent Wright County.

heavy summer rains brought the 1990 total to 77.5 cm (U.S. Weather Service, 1991).

2.2. Wright County, Minnesota Wetlands

Wright County is located to the west of the TCMA, at the eastern fringe of the prairie pothole region (van der Valk, 1989). It has an area of 1850 km^2, 90% of which was in crop production in the 1960's (Edwards, 1968). All of Wright County was once glaciated, and as a result, the topography consists of outwash plains and rolling hills, with marshes in many of the glacial depressions (Edwards, 1968). Wright County wetlands are usually hydrologically isolated with respect to other surface waters, and receive recharge from precipitation and groundwater flow.

2.3. Study Background

Data for analysis of temporal and spatial variability of wetland water quality were collected during pre-treatment phases of two studies in the TCMA and adja-

TABLE I

Location and timing of studies in Twin Cities Metropolitan Area (TCMA), Minnesota (USA) used to evaluate and spatial variability in wetland water-quality

Location of study	Time period	Description of wetlands	Purpose of comparison	Specific statistical test(s)
Seven-county TCMA	Snowmelt 1989 Spring-fall 1989	39 wetlands	Analysis of categorical factors influencing wetland water-quality variability	ANOVA or Kruskal-Wallis, Tukey–Kramer
Wright County	Spring 1989, 1990, and 1991	18 marshes	Analysis of interannual variability	ANOVA, Paired t-tests
Wright County	Snowmelt 1989–spring 1991	18 marshes	Analysis of power of completely randomized design vs. before-and-after design	
Wright County	Mid-July 1990	Transects within 8 marshes	Comparison of individual sample measurements vs. composite sample measurements	ANOVA or Kruskal-Wallis, Paired t-test
Wetland 19–3–2, Wright County	Mid-July 1990	9.5 ha deep semi-permanent march	Geostatistical analysis Regression analysis of intra-wetland factors affecting spatial variability in water quality	
Wetland 6–3–3, Wright County	Mid-July 1990	9.5 ha shallow seasonal march	Geostatistical analysis Regression analysis	

cent Wright County (Table I). Selection criteria and individual site characteristics are described in Detenbeck *et al.* (1992). All forty-nine study-site wetlands were categorized according to predominant surrounding land-use, hydrologic regime (isolated basins, intermittent surface water flow, or continuous surface water flow-through), and wetland type by Circular 39 classifications (Shaw and Fredine, 1956).

3. Methods

3.1. Seasonal Sampling

Thirty-one wetlands in the Twin Cities metropolitan area and eighteen wetlands in adjacent Wright County were sampled over four seasons of the predisturbance (or pre-treatment) phase in 1989: snowmelt (mid March to mid April), late spring (mid May to mid June), summer (mid July to mid August), and early fall (mid September to mid October; Table I). Sampling was stratified by wetland type, based on Circular 39 classifications (Shaw and Fredine, 1956). Open bodies of water within wetlands were sampled as far from shore as possible, from mid-depth in the water column. To obtain a more representative sample, three to five grab samples were composited for each sampling site. Wetland types characterized by shallow (<10 cm), patchy, or nonexistent standing water were sampled along established groundwater well transects. Samples from transects were pooled according to wetland type before analysis, except as described below for studies of spatial variability within wetlands.

Sampling wells were constructed from 30 cm lengths of 3.75 cm diameter schedule 40 PVC pipe, with 1 mm width slots. PVC were washed in non-phosphate detergent to romave mold-release lubricant from the surface, and then soaked in distilled water for at least 1 h. Pipes were capped at the bottom with a PVC cap and at the top with a rubber stopper to keep sediment and precipitation out.

Samples were collected from PVC wells with a distilled-water washed baster. Pipe or surface water composites were stored at 4 °C, separated and filtered for nutrient analysis (0.45 μ) within 12 h of collection, and then preserved by freezing (nutrients), acidification with H_3PO_4 (organic carbon) or HNO_3 (total extractable lead), or cooling to 4 °C (turbidity, specific conductivity, suspended solids, color). Ultra-pure (Baker Instra-analyzed*) nitric acid was used in acidifying metal samples. Samples collected from PVC wells or at sites where sampling disturbed underlying sediments were analyzed for dissolved constituents only.

Water temperature, dissolved oxygen (YSI Model 54A Meter), surface or water table depth (within pipes), and long-term water table depth were measured in the field. The rusty-rod technique (Hook *et al.*, 1987) was used as a time-integrated measure of water table depth.

3.2. Spatial Sampling

The scale of spatial variability was examined by sampling one shallow and one deep cattail marsh in Wright County very intensively (Table I). Each wetland was gridded into 50 × 50 meter cells on a U.S.G.S. $7\frac{1}{2}$ minute topographic map. Coordinates of 28–30 sample locations were selected using a random number table. Sample locations were located on the ground by orienteering with a compass, using measured paces from a known reference point. At each sample point, water depth, temperature, and dissolved oxygen were measured. Dominant vegetation class ((*Typha* spp.), willow (*Salix* spp.), grasses/sedges, reed canary grass (*Phalaris arundinaceae*), open water) and an index of shading or plant density (full sun, partly shaded, completely shaded) were recorded at each site. Vegetation classes were ranked roughly according to the dominance of facultative vs. obligate wetland species (1 = cattail, 2 = grasses/sedge, 3 = reed canary grass, 4 = willow/alder).

The effect of compositing transect subsamples on estimates of wetland water quality was tested by sampling within four shallow and four deep cattail marshes in Wright County. Transects were established in a random direction starting from

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a point of ready access. Within each wetland, five transect stations were marked at intervals of approximately 10 m. A separate set of subsamples was collected at each established transect station. In addition, equal volumes from each station along a given transect were composited into a single container for analysis.

3.3. Analytical Methods

Samples were analyzed according to Standard Methods (APHA, 1985) or by EPA-approved procedures except as noted (Table II). Field blanks, lab replicates, field replicates, quality control (QC) standards, and spiked samples were analyzed for 10% of samples to determine detection limits, precision, accuracy, and percent recovery (Table II).

3.4. Statistical Tests

In all of the following analyses, data were tested for normality using the Wilke-Shapiro test (SAS 1988) and for homogeneity of variance by Bartlett's test (Snedecor and Cochran 1980). Where necessary, $\log_e$ transformations were performed on data prior to analysis. When transformations were not sufficient to normalize data or homogenize variances, an equivalent nonparametric (Kruskal–Wallis) test was used.

Analysis of variance (ANOVA) or paired t-tests were used to analyze (a) effects of intrinsic and extrinsic factors on pre-disturbance wetland water quality; (b) effect of year on spring pre-disturbance wetland surface-water and groundwater quality; and (c) effects of compositing samples on water quality measures (SAS, 1988; Table I). For the first category of tests, separate tests were performed for 1989 snowmelt data and for data averaged over the growing season (mid May–mid October). Test for differences among categories were run using the Tukey Kramer test for multiple comparisons (SAS, 1988).

The power of completely randomized designs vs. paired comparisons (before-and-after) designs were compared after $\log_e$-transforming data from Wright County sites. First, for each season (spring, summer), the change in $\log_e$-transformed data between pre- and post-treatment years was compared between control and methoprene or BTI treatment sites. Second, for 1991 data only, differences between May pretreatment data and data for each post-treatment date were computed and compared between control and treatment sites. For a representative set of each of the above comparisons, the power of detecting a 100% difference between treatments and controls was calculated (Hodges and Lehmann, 1968). In the case of before-and-after paired comparisons, the probability of detecting an *additional* increase of 100% beyond that due to climate change alone was calculated. Pooled means and variances were used in calculating the power of statistical tests.

For multiple samples within a wetland, single or multiple regression analyses were performed to analyze the simple effects of water depth, dissolved oxygen,

TABLE II

Methods and quality assurance data for water quality analyses

Parameter	Method	Instrument	Ref.[a]	Detection limit[b]	Lab precision[c]	Field precision[c]	Accuracy[d]	Percent Recovery[d]
Turbidity	Nephelometric	DRT turbidimeter	1	1.0 NTU	3	24	–	–
Total suspended solids	Gravimetric	–	1	0.005 g/l	15	25	–	–
Volatile suspended solids		–				35		
Specific conductivity	–	Barnstead PM70CB	1	11 μmhos cm^{-1}	0.8	2	–	–
Color	Spectrophotometric	–	2	5.2 PCU	3	4	–	–
Ammonium	Salicylate-hypochlorite	Latchet	3	0.02 mg N/l	6	19	100±3	102±18
$NO_3 + NO_2$	Cadmium reduction	Latchet	3	0.02 mg N/l	3	23	94±1	96±13
Dissolved N	Digestion/	Latchet	3,4	0.1 mg N/l	4	5	95±3	98±6
Total N	Cadmium reduction			0.04 mg N/l		9		97±25
Soluble reactive P	Ascorbic acid, bisulfite pretreat	Latchet	3,5	0.009 mg P/l	3	7	96±7	104±18
Dissolved P	Digestion/	Latchet	3,4	0.01 mg P/l	4	6	99±3	99±10
Total P	ascorbic acid method			0.004 mg P/l		11		96±20
Total extractable Pb	Digestion, atomic absorption spectrometry	Perkin–Elmer Zeeman 5100	5,6	0.003 mg Pb/l	17	34	89±22	89±44
Dissolved organic C	Oxidation/IR	Dohrman C	5	2.9 mg C/l	2	4	95±3	-
Total organic C		analyzer		2.2 mg C/l				

[a] References: 1 = APHA, 1985; 2 = Buffle *et al.* 1982; 3 = Latchet, 1989; 4 = Solorzano and Sharp, 1980; 5 = US EPA, 1983; 6 = Gilman, 1988.
[b] Defined as 2 s.d. of field blanks.
[c] Relative standard deviation (% of mean).
[d] Percent of expected (mean ± s.d.).

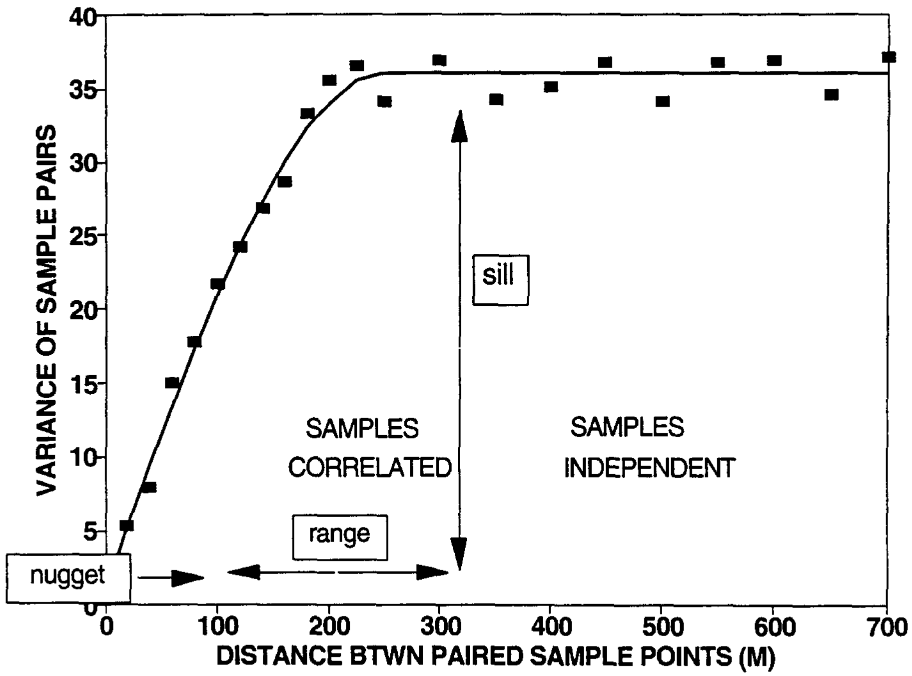

Fig. 2. Sample variogram. The nugget (1) is the estimated interpair variability for an interpair distance of zero. The sill (35) is related to the variability among samples not affected by spatial autocorrelation, and the range (250 m) is the intrapair distance at which the variogram levels off.

water temperature, distance from the wetland edge, and location within the wetland on water quality variables. North-south coordinates measured the distance from the road located along one edge of each wetland, while east-west coordinates measured the distance between upland areas with differing land-use on either side of each wetland. Canonical correlation analysis was applied to each wetland separately to analyze multiple correlation between site characteristics and water quality variables (SAS, 1988).

Geostatistical analyses were applied to determine the relative scale of spatial variability for each water quality variable measured. A variogram was constructed for each variable using the GEOEAS software package (Englund and Sparks, 1988). In a variogram, the average variance between sample pairs is plotted as a function of distance between sample pairs (Figure 2). Each variogram is characterized by a nugget, a range, and a sill (Knudsen, 1988; Figure 2). The nugget corresponds to the estimated variability at a distance of zero, and is related to sample collection and analytical error. The range corresponds to the intrapair distance at which the variogram levels off. The range is a measure of the 'grain' of the wetland (Forman and Godron, 1986), i.e. the distance over which water quality is relatively homogeneous.

TABLE III

Theoretical power of detecting treatment effects on June 1990 or June 1991 water quality values or on interannual (1991–1990) or interseasonal (June 1991–May 1991, Sept 1991–May 1991) differences in midwetland water quality. Calculations are based on an assumed experimental design with 8 control wetlands and 4 wetlands treated with *Bacillus thuringiensis* in Wright County, MN

	Probability to detect a 100% difference		Probability to detect an additional 100% increase between years	Probability to detect an additional 100% increase between seasons	
Variable	June 1990	June 1991	June 1991/ June 1990	June 1991/ May 1991	Sept 1991/ May 1991
H_2O depth	0.94	0.76	0.35	0.91	0.34
Nitrate	0.10	0.08	0.23	0.12	0.18
Ammonium	0.22	0.20	0.13	0.06	0.09
Dissolved N	0.89	0.86	0.97	0.26	0.27
Total N	0.54	0.36	0.79	0.12	0.19
OrthoPO_4	0.59	0.16	0.25	0.13	0.20
Dissolved P	0.55	0.19	0.27	0.09	0.32
Total P	0.53	0.11	0.65	0.05	0.08
Dissolved organic C	0.97	0.84	0.92	0.70	0.60
Total organic C	0.97	0.31	0.98	0.18	0.38
Total suspended solids	0.21	0.14	0.07	0.03	0.03
Volatile suspended solids	0.23	0.15	0.08	0.03	0.04
Specific conductivity	0.94	0.92	0.99	0.74	0.49
Turbidity	0.26	0.16	0.10	0.03	0.03
Color	0.80	0.45	0.75	0.43	0.47
pH	–	1.00	–	1.00	1.00
Alkalinity	–	0.94	–	0.65	0.29
Dissolved oxygen	–	0.29	0.88	1.00	0.62

4. Results

4.1. Power of Tests for Completely Randomized vs. Paired Comparisons Designs

To facilitate comparisons of different experimental designs, the power of analyses was recalculated using a standard number of controls (n = 8) and treatment sites ($n = 4$; Table III). For both interannual and interseasonal paired comparison designs, detection of ecologically significant effects probably would be limited by the power of our analyses (Table III). The power for detecting significant effects on soluble reactive phosphate, nitrate, ammonium, total dissolved phosphorus, turbidity, and total or volatile suspended solids was very low (3–25%). The probability of detecting significant effects on interannual changes was high (92–99%) for specific conductivity, total dissolved N, and dissolved or total organic C, moderately high (65–88%) for total N, total P, color, dissolved oxygen, and much lower for other variables. The probability of detecting treatment effects on interseasonal change was moderately high to high for dissolved oxygen, pH, color, specific conductivity, and dissolved organic carbon (43–100%; Table III), but low for other variables. For many variables, the calculated power of tests conducted within a single year varied widely between 1990 and 1991.

Overall, the power of experimental designs for detecting a 100% difference in treatment vs control means within June 1990 was roughly comparable to the power associated with detecting a 100% increase in interannual change between June 1990 and June 1991. However, the power of detecting differences among sites within a year was not consistent between 1990 and 1991 for total N, soluble reactive P, total dissolved P, total P, total organic C, and color. Thus, the before-and-after paired comparison design is probably the most powerful design overall.

4.2. Sources of Background Variation in Wetland Water Quality

Within a given year, pre-disturbance wetland water quality was significantly affected by contact with the sediment (surface water vs. groundwater), season, degree of hydrologic isolation, wetland class, and predominant land-use in the surrounding watershed ($p < 0.05$; Table IV–VI). Nitrate + nitrite, ammonium, and total dissolved N were two to fifteen times higher during snowmelt periods, while total organic C was lower during snowmelt than during the spring sampling season ($p < 0.05$).

Mean groundwater concentrations were significantly higher than surface water concentrations ($p < 0.05$) for total dissolved N and color. Over the growing season (spring–fall), shallow marshes had higher concentrations of nutrients, organic carbon, and specific conductivity than wetland ponds. Soluble reactive phosphate and total dissolved phosphorus differed significantly among all classes tested: shallow marshes, deep marshes, ponds, and wet meadows + seasonally-flooded systems (Table IV). In general, wetlands in isolated basins had significantly higher nutrient, dissolved organic C, and color values than wetlands with intermittent or continuous flow (Table V).

Agricultural land-use in the surrounding watershed had a significant effect on wetland water quality as compared to urban/residential land-use or relatively undeveloped zones (Table VI). During snowmelt, particulate nitrogen, dissolved organic C, and specific conductivity were significantly higher in midwetland samples from sites surrounded by agricultural lands as compared to sites surrounded

TABLE IV

Effect of wetland type by Shaw and Fredine (1956) classification on midwetland surface water quality at 39 wetlands in the Minneapolis/St. Paul metropolitan area and adjacent Wright County during 1889 growing season

Water quality variable	Relationship of water quality by wetland type (geometric class mean)	Significance
Total nitrogen, mg N/l	SM > PD (3.23) (1.12)	[a]
Total phosphorus, μg P/l	SM > PD (677) (68)	[a]
Dissolved phosphorus, μg P/l	SM > DM > SS, WD, WM/SF > PD (173) (85) (37) (16)	[a]
Soluble reactive phosphorus, μg P/l	SM > DM > SS, WD, WM/SF > PD (91) (39) (17) (3)	[a]
Dissolved organic carbon, mg C/l	SM, SM > PD (22.6, 20.9) (14.2)	[b]
Specific conductivity, μmho's/cm	SM > PD, DM (530) (467, 305)	[b]
Color, PCU	SM > PD (201) (73)	[a]

(SM, shallow marsh (Type 3); PD, pond (Types 5); DM, deep marsh (Type 4); SS, scrub-shrub wetland (Type 6); WD, wooded swamp (Type 7); WM/SF, wet meadow/seasonally-flooded wetlands (Types 1 and 2).

[a] $p < 0.05$.

[b] $p < 0.01$.

by urban/residential or undeveloped lands. During the growing season, ammonium, total dissolved N, soluble reactive phosphate, total dissolved P, particulate P, dissolved organic C, color, suspended solids, and specific conductivity were significantly higher within wetlands surrounded by agriculture than within those surrounded by other land-use types.

When data for all Wright County wetlands sampled were pooled, nitrate plus nitrite increased significantly between 1989 and 1990 (diff = 417 μg NO_3NO_2-N/l; paired t-test, $p < 0.01$), but no other water quality variable changed significantly. Surface water nitrate plus nitrite and particulate N increased significantly between years (paired t-test, increase = 48 μg NO_3+NO_2-N/l. $p < 0.01$; 6 mg PN/l, $p < 0.05$). Total dissolved N in groundwater decreased significantly between years (diff. = 2 mg N/l, $p < 0.05$). At these same sites, the level of the time-integrated water table increased by 8 cm between years ($p < 0.05$). Within 1989 groundwater/1990 surface water pair comparisons, only specific conductivity changed significantly between years, decreasing by 290 μmhos cm^{-1} ($p < 0.05$).

TABLE V

Effect of wetland hydrologic class of 39 wetlands in Minneapolis/St. Paul metropolitan area and adjacent Wright County on pre-disturbance wetland water quality

Water quality variable	Relationship during snowmelt period[a] (geometric mean)	Significance[b]	Relationship during spring – fall period[c,d] (geometric mean)	Significance[b]
Dissolved nitrogen, mg N/l	–	–	IS > $\overline{\text{INT, FL}}$ (2.15) (1.34, 0.93)	**
Total phosphorus, μg P/l	IS > INT (585) (311)	(*)	$\overline{\text{IS} > \text{FL}}$ > $\overline{\text{INT}}$ (506) (280) (168)	**
Dissolved phosphorus, μg P/l	–	–	IS > $\overline{\text{INT, FL}}$ (157) (42, 32)	**
Soluble reactive P, μg P/l	–	–	IS > $\overline{\text{FL, INT}}$ (98) (15, 8)	**
Dissolved organic C, μg P/l	IS > INT (17.1) (14.5)	*	$\overline{\text{IS, INT}}$ FL (21.2, 17.6) (12.7)	*
Color, PCU	–	–	$\overline{\text{IS} > \text{IN}}\overline{\text{T} > \text{FL}}$ (185) (96) (100)	**

IS, isolated basin without inflows or outflows; INT, intermittent flow; FL:, flow-through with inlet(s) and outlet.

[a] snowmelt period = mid March–mid April.

[b] * $p < 0.05$; ** $p < 0.01$ by one-way ANOVA, d.f. corrected for unequal n; (*) $p < 0.05$ by Kruskal-Wallis test to correct for unequal variances.

[c] spring-fall = mid May – mid October.

[d] Bar overlying groups indicates no significant difference between groups by Tukey's multiple comparison of means test.

4.3. Spatial Variability in Water Quality within Wetlands

For the two wetlands sampled intensively, soluble reactive phosphate and total dissolved P were the most spatially variable (coefficient of variance = 76–249%) while temperature, color, dissolved organic carbon, and dissolved oxygen were the least variable (c.v. = 6–43%; Table VII). The average distance across which water quality variables were spatially correlated (variogram range) was between 61 and 112% of the mean radius of each wetland (Table VIII):

$$\text{Site } 6-3-3: \text{ Avg. variogram range} = 174 \text{ m } (60-300 \text{ m}),$$
$$\text{Avg. radius} = 155 \text{ m}$$

TABLE VI

Effect of predominant surrounding land-use type on pre-disturbance midwetland water quality of 39 wetlands in the Minneapolis/St. Paul metropolitan area and adjacent Wright County, 1988–1989

Water quality variable	Differences during snowmelt period[a] (geometric mean)	Significance[b]	Differences during spring-fall period[c,d] (geometric mean)	Significance[b]
Ammonium μg N/l	–	–	AG > UN > U/R (62) (39) (59)	*
Dissolved nitrogen mg N/l	–		AG > UN > U/R (2.25) (1.44) (1.63)	**
Particulate nitrogen mg N/l	AG > U/R (0.45) (0.30)	*	–	–
Soluble reactive phosphorus, μg P/l	–	–	AG > $\overline{\text{U/R, UN}}$ (188) (27, 15)	**
Dissolved phosphorus, μg P/l	–	–	AG > $\overline{\text{U/R, UN}}$ (306) (77, 36)	**
Particulate phosphorus, μg P/l	–	–	$\overline{\text{AG, UN}}$ > U/R (231, 194) (134)	*
Total phosphorus, μg P/l	–	–	AG > $\overline{\text{U/R, UN}}$ (822) (247, 207)	**
Dissolved organic carbon mg C/l	AG > UR (19.7) (14.5)	*	AG > U/R > UN (36.1) (15.1)) (14.6)	**
Color, PCU	–	–	AG > $\overline{\text{U/R, N}}$ (256) (119, 100)	**
Total suspended solids, mg/l	–	–	AG > U/R > UN (167) (35) (35)	*
Volatile suspended solids, mg/l	–	–	AG >$\overline{\text{UN, U/R}}$ (97) (24, 17)	**
Specific conductivity, μmhos/cm	AG > U/R (392) (150)	*	AG > UN > U/R (601) (419) (291)	(*)

[a] Snowmelt period = mid March–mid April.
[b] * $p < 0.05$; ** $p < 0.01$ by one-way ANOVA, d.f. corrected for unequal n; (*) $p < 0.05$ by Kruskal-Wallis test to correct for unequal variances.
[c] spring–fall = mid May–mid October.
[d] Bar overlying groups indicates no significant difference between groups by Tukey's multiple comparison of means test.
AG, agricultural; UN, undeveloped; U/R, urbanan/residential.

TABLE VII

Summary of water-quality data (mean, range, and coefficient of variation) for deep marsh (site 19–3–2) and shallow marsh (site 6–3–3) in Wright County, MN in summer 1990

Variable	Deep marsh *n*	Deep marsh Mean (range)	Deep marsh C.V. (%)	Shallow marsh *n*	Shallow marsh Mean (range)	Shallow marsh C.V. (%)
Soluble reactive P, μg P/l	28	189 (2.5–770)	105	30	108 (3–1321)	249
Dissolved P, μg P/l	28	345 (99–999)	76	30	346 (60–2031)	127
Ammonium, μg N/l	28	418 (4–3870)	232	30	61 (4–196)	68
Nitrate, μg N/l	28	54 (20–152)	63	30	62 (21–117)	33
Dissolved N, mg N/l	28	2.37 (1.15–7.21)	65	30	4.00 (1.88–8.25)	41
Dissolved organic C, mg C/l	27	35.4 (19.4–64.7)	36	30	72 (36–112)	28
Color, PCU	27	285 (97–473)	36	30	292 (60–552)	35
Specific conductivity, μmhos/cm	27	486 (221–1050)	50	30	160 (65–573)	64
Dissolved oxygen, mg/l	27	4.16 (0.40–7.90)	44	30	3.72 (2.00–6.90)	35
Temperature, °C	28	24.5 (22.4–29.0)	6	30	24.0 (20.7–29.5)	8
Water depth, cm	28	9 (1–24)	64	30	12 5–18	27

$$\text{Site } 19-3-2: \text{ Avg. variogram range } = 120 \text{ m } (60-200), \text{ Avg. radius } = 194\text{m}$$

In the shallow marsh, 6–3–3, ammonium exhibited the smallest-scale patchiness (Figure 3a), similar to that of the sun/shade index (ranges = 60–80 m; Table VIII). In the deep marsh, 19–3–2, dissolved oxygen, soluble reactive phosphate, and total dissolved N exhibited small-scale patchiness, similar to that of water depth (Table VIII, Figure 3b).

Canonical correlations were not significant for site 19–3–2, but were significantly greater than zero for site 6–3–3 (H_0: All CC = 0, Wilke-Lambda test, $p <$

TABLE VIII

Parameters for fitting variograms to spatial water quality data for two Wright County, MN cattail marshes, summer 1990

Variables	Wetland 6–3–3 (average radius = 155 m)				Wetland 19–3–2 (average radius = 194 m)			
	Model[a]	Nugget	Range (m)	Sill	Model[a]	Nugget	Range (m)	Sill
Water depth	S	0	140	11	S	0	100	35
Vegetation[b]	S	0.02	140	0.14	S	1	260	0.5
Sunlight[c]	S	0	80	0.65	S	0.1	160	0.35
Dissolved oxygen	S	0.20	100	1.75	S	0	60	3.7
ln Soluble reactive P	S	0	180	3.70	S	0	80	1.8
ln Dissolved P	S	0	300	0.85	S	0	100	0.5
ln Ammonium	S	0	60	0.65	S	0	100	4.2
ln Nitrate + nitrite	S	0.03	160	0.08	S	0.15	160	0.22
ln Dissolved N	S	0	160	0.16	G	0	80	0.27
Dissolved organic cabon	G	100	250	500	G	0	180	240
Color	S	0	20	10000	S	4000	200	8000
In Specific conductivity	S	0.15	160	0.15	E	0	120	0.22
Temperature					S	0.5	140	2
Average			174				120	

[a] Model types: S, spherical; E, exponential; G, Gaussian.
[b] Index of facultative (1) to obligate (5) wetland-dependence.
[c] Index of shading (1–5).

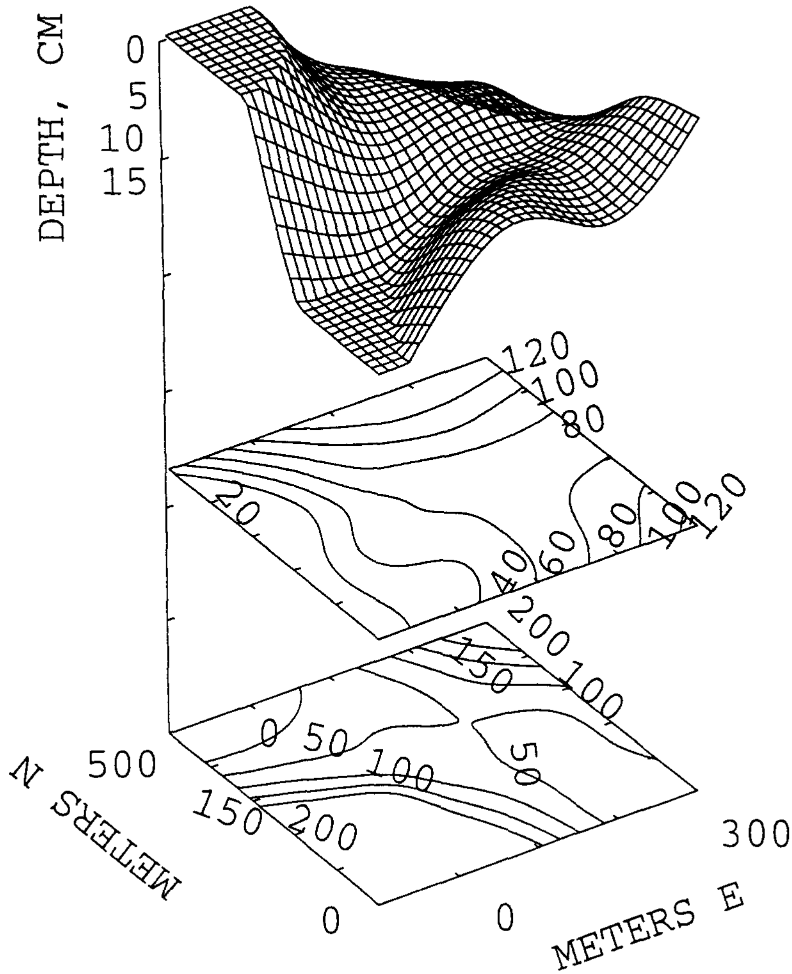

Fig. 3a.

0.05). Variation in water quality within wetland 19–3–2 was more readily expressed through simple regression analysis with depth or distance from edge as predictor variables than through CCA (see below). For site 6–3–3, the first four canonical correlations (CC) explained 96% of the variance. The first canonical correlation (CC1) expressed the association of high dissolved oxygen, facultative/obligate wetland vegetation (FAC/OBL; reed canary grass or scrub/shrub), and high light intensity with high specific conductivity and high total dissolved N but low ammonium. The second CC expressed the association of high dissolved oxygen, shallow water depth, and the EW gradient with low nitrate + nitrite or conductivity and high dissolved organic carbon. The third CC expressed the association of the NS distance and FAC/OBL vegetation with a combination of high nitrate + nitrite and total dissolved nitrogen. Finally, the CC4 expressed the 'edge' effect. Sites near the edge with shallow water depth, more intense sunlight, higher temperature, and

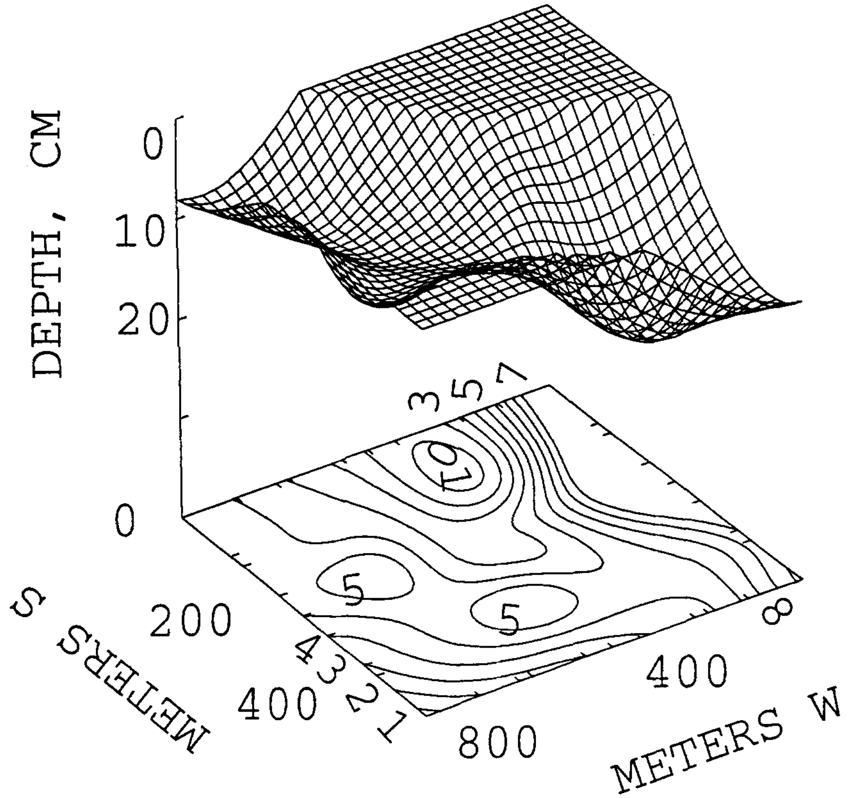

Fig. 3b.

Figs. 3(a)–(b). Spatial variability in water quality variables in Wright County, MN cattail marshes. (a) Marsh 6–3–3, from top to bottom: 3-D map of depth (cm), contour map of dissolved organic carbon (mg C/l), and contour map of ammonium (μg N/l). (b) Marsh 19–3–2, from top to bottom: 3-D map of depth (cm) and contour map of dissolved oxygen (mg/l).

FAC/OBL vegetation tended to have higher specific conductivity, soluble reactive phosphate, and total dissolved P levels (Table IX).

Within the shallow cattail marsh (6–3–3) only specific conductivity decreased exponentially with distance from the wetland edge (Table X. $P < 0.001$). Dissolved organic carbon and total dissolved P decreased linearly with depth. Color increased with DO up to 4 mg O_2/l, where it approached an asymptote of approximately 400 PCU. Dissolved organic carbon increased in a westerly direction along the EW gradient (Table X).

Nitrogen speciation within wetland 6–3–3 varied with dissolved oxygen. Both ammonium and the fraction of total dissolved nitrogen in mineralized form (NH_4+NO_3+NO_2) decreased as a function of dissolved oxygen in the range of 0–4 mg O_2/l ($p < 0.001$; Table X). Conversely, the fraction of mineralized N

TABLE IX

Significant canonical correlations with water quality variables and site characteristics for a shallow marsh (site 6–3–3) in Wright County, MN ($p < 0.05$)

Variable	CCC1	CCC2	CCC3	CCC4
Water quality				
Color	–	–	–	–
Ln specific conductivity	0.485	–0.433	–	0.652
Nitrate + nitrite	–	–0.453	0.534	–
Ln ammonium	–0.714	–	–	–
Ln dissolved nitrogen	0.569	–	0.714	–
Ln soluble reactive phosphorus	–	–	–	0.608
Ln dissolved phosphorus	–	–	–	0.651
Ln dissolved organic carbon	–	0.420	–	–
Site variables				
Dissolved oxygen	0.594	0.425	–	–
Temperature	–	–	–	0.619
NS coordinate	–	–	0.682	–
EW coordinate	–	0.700	–	–
Distance from edge	–	–	–	–0.466
Water depth	–	–0.643		–0.466
Vegetation index	0.526	–	0.428	0.405
Sunlight index	0.438	–	–	0.431

present as nitrate + nitrite increased as a function of dissolved oxygen over this same range ($p < 0.05$).

Within the deep marsh (19–3–2), most water quality variables were explained as a simple function of water depth or distance from edge (Table X). Color, dissolved organic carbon, nitrate + nitrite, ammonium, total dissolved nitrogen, and the fraction of dissolved nitrogen in mineralized form all decreased linearly as a function of water depth, while total dissolved phosphorus and the fraction of mineralized N as nitrate + nitrite increased as a function of depth. Specific conductivity decreased with distance from wetland edge. Only temperature varied with the NS and EW gradient, increasing towards one corner of the wetland.

Measurement of most water quality variables was not biased by compositing samples prior to analysis. Only ammonium differed significantly between composite and transect average values ($p < 0.05$). Transect average values for ammonium were greater (170 μg NH_4-N/l) than sample composite values. Ammonium also was the most variable within transects (c.v. = 52–147%). Only specific conductivity varied significantly between transect stations (Kruskal-Wallis test, $p < 0.05$).

TABLE X

Multiple regressions explaining spatial variability in water quality data within two Wright County, MN cattail marshes. The last two regressions are valid only within the range of 0–4 mg O_2/l

Site	Regression equation	Adj'd r^2
19–3–2	Temperature (°C) = 21 + 0.007 EW coord + 0.025 NS coord	0.37^b
	$\log_e$ Specific conductivity (μmhos/cm) = 6.3 – 0.015 (distance from edge, m)	0.23^a
	Color (PCU) = 346 – 6.8 (depth, cm)	0.15^a
	$\log_e$ DOC (mg C/l) = 3.87 – 0.038 (depth, cm)	0.43^c
	$\log_e$ Total dissolved P (μg P/l) = 5.2 + 0.05 (depth, cm)	0.11^a
	$\log_e$ NH_4 (mg N/l) = 5.95 – 0.020 (depth, cm)	0.36^c
	$\log_e$ Total dissolved N (mg N/l) = 1.2 – 0.06 (depth, cm)	0.45^c
	(NO_3-N + NO_2-N)/(total inorganic N) = 0.25 + 0.023 (depth, cm)	0.24^b
	$\log_e$ Inorganic N/Total dissolved N = 6.2 – 0.9 ($\log_e$ [depth, cm])	0.49^c
6–3–3	$\log_e$ Specific conductivity (μmhos/cm) = 567 – 59 ($\log_e$ [distance from edge, m]	0.65^b
	$\log_e$ Color (PCU) = 3.6 + 1.0 (D.O., mg/l) – 0.1(D.O)^2	0.24^b
	Dissolved organic carbon (mg C/l) = 109 – 2.8 (depth, cm)	0.20^b
	DOC (mg C/l) = 100 + 0.011 (EW coord) – 0.05 (NS coord)	0.68^c
	$\log_e$ Soluble reactive P/Dissolved P = 0.5 – 0.05 ($\log_e$ [EW coord])	0.18^a
	$\log_e$ Dissolved P (mg P/L) = 6.6 – 0.1 (depth, cm)	0.15^a
	$\log_e$ Ammonium (mg N/L) = 5.5 – 1.3 (D.O., mg/l)	0.33^b
	(NO_3-N + NO_2-N)/(total inorganic N) = 0.04 + 0.16 (D.O., mg/l)	0.39^a
	Inorganic N/Total dissolved N = 0.10 – 0.020 (D.O., mg/l)	0.50^c

Significance levels: [a] $p < 0.05$; [b] $p < 0.01$; [c] $p < 0.001$.

Specific conductivity was highest relative to the mean (average ratio = 1.17) at the transect station nearest the edge and decreased gradually with distance from the edge to the innermost point on the transect (average ratio = 0.93).

5. Discussion

5.1. Effect of Temporal vs Geographic Variability on Power of Tests for Different Experimental Designs

Water-quality variables and associated physical measurements can be classified based on their relative control by climatic, edaphic, land-use, or biotic factors (Magnuson *et al.*, 1990). Variables controlled primarily by climate include water depth and dissolved oxygen, while those controlled by edaphic factors include alkalinity, pH and specific conductivity. Variables controlled by edaphic factors in pristine landscapes, but by land-use within developed watersheds include total nitrogen and total phosphorus. Variables heavily influenced by biotic interactions (e.g. bacterial activity) include nitrate + nitrite, ammonium, total dissolved nitrogen, dissolved organic carbon, soluble reactive *P*, total dissolved *P*, and volatile suspended solids. Water quality variables influenced mainly by biotic interactions can be further broken down into those dependent on redox conditions and thus climatic/biotic interactions (i.e. nitrate + nitrite, ammonium, orthophosphate) and those tied to processing of organic matter (dissolved organic carbon, total dissolved nitrogen). Finally, some variables will be strongly influenced by more than one factor, e.g. suspended solids and turbidity (climatic, biotic factors) and color (biotic, edaphic factors).

The probability of detecting treatment effects of 100% for variables primarily related to climatic or edaphic factors was relatively high, particularly for between-site comparisons or within-site/between year comparisons (power = 86–94%). Change in water depth or dissolved oxygen was actually more predictable within sites and between adjacent seasons than within-sites, between years. Water-quality variables related to the detritivore food chain (dissolved organic carbon, total organic carbon, total dissolved nitrogen) also were highly predictable among sites within years or within sites between years, but not within sites, between seasons. Water quality variables influenced by external land-use (total phosphorus, total nitrogen) were moderately predictable within sites, between years (power = 65–79%).

The least predictable water quality variables were those influenced by a combination of biotic factors and redox conditions (NO_3+NO_2, NH_4), by a combination of biotic, climatic and/or edaphic conditions (turbidity, TSS, color) or by complex biotic interactions (volatile suspended solids). The power of detecting a 100% difference or a 100% change beyond background temporal variability for all of these variables except color was extremely low for all experimental designs tested (power = 3–59%). Color is highly correlated to dissolved organic carbon, but also is affected by pH and the relative proportion of humic vs. tannin materials (Buffle *et al.*, 1982); thus the power of detecting color changes between sites or over time was slightly less than that for dissolved organic carbon (power = 75–80%).

5.2. Sources of Variability in Wetland Water Quality

Little work has been done previously to quantify the impact of different land-use types on wetland water quality. Within Eagan, which is rapidly being converted from undeveloped and low-intensity agricultural land to suburban developments, there is a linear relationship between percent urban area in watersheds and total phosphorus levels in lakes and wetland ponds (Ayers *et al.*, 1980). Over the wider TCMA, lake trophic status over a 30-year period has been explained as a function of agricultural land-use, and the coverage, type, and spatial distribution of wetlands within watersheds (Detenbeck *et al.*, 1993). Lakes surrounded by predominantly agricultural land-use were relatively more eutrophic than those surrounded by other land-uses. These findings are consistent with our results for wetland water quality.

The most marked seasonal effects on wetland water quality observed in our study were significantly higher levels of total dissolved N forms and lower total organic carbon concentrations measured during snowmelt as compared to the spring sampling period. In agricultural watersheds of the TCMA, loadings of most nutrients and suspended sediments are highest during spring runoff when infiltration is minimal (Oberts and Jousseau, 1979). Likewise, in an urban wetland in St. Paul, the majority of inorganic suspended solids, nitrate plus nitrite, and ammonium loadings occurred during March and April (Brown, 1985b). Biological processes that consume or transform dissolved N (nitrification, denitrification, ammonification, plant uptake) are all temperature-dependent (Q_{10} = 2.6–2.9; Bowden, 1987; Westermann and Ahring, 1987), and operate slowly during the snowmelt period. In two wetlands receiving agricultural runoff in western Minnesota, the most rapid increases in N content of decomposing cattail litter and above-ground biomass occurred in late May and early June, respectively (Vanamburg, 1982).

Most snowmelt and precipitation is lost as runoff when the wetland sediments are still frozen, so that contact with the sediments in minimal during early spring. Thus, difusion-limited processes such as denitrification will be inhibited by short retention times (Brown, 1985a; Bowden, 1987). In studying the cumulative function of wetlands within stream drainages of the Twin Cities metropolitan area, Johnston *et al.* (1990) found that wetlands were least effective in removing nitrate during high flow conditions. In contrast to dissolved inorganic N loadings, total organic N and volatile suspended solids loadings may remain high well into the growing season in both urban (Brown. 1985a) and agricultural wetlands (Oberts and Jousseau, 1979).

Our findings with respect to higher groundwater concentrations of ammonium and total dissolved nitrogen relative to concentrations in surface water were consistent with Kadlec's (1986) comparisons. Kadlec (1986) found significant variation in surface water quality among experimental wetland cells in a Manitoba marsh. Groundwater concentrations of soluble reactive phosphate and ammonium-N were significantly higher than surface water concentrations, and were relatively less variable, both among wetland cells and among seasons. The interaction between wetland cell and season was not significant, suggesting that wetland groundwater quality was temporally coherent among cells of this wetland complex. Unlike Kadlec, we found no significant differences between groundwater and surface water soluble reactive phosphate measurements.

Few data are available comparing water quality of different wetland types, but some differences can be expected as the result of varying sedimentation efficiencies. Walker (1987) explained 89% of the variation in total phosphorus retention by urban stormwater retention ponds or wetlands with a model based on (a) total phosphorus loadings proportional to impervious surface area in the watershed, (b) retention time as a function of the wetland volume : outflow ratio, and (c) 2nd order sedimentation kinetics. Thus, wetlands with greater volume (capacity) will tend to have a longer water retention time during which sedimentation of particles and particulate nutrients can occur. Within a range of marshes, ponds, and lake systems of the TCMA, Osgood (1988) found the ration of mean depth : square root (surface area) to be a good predictor of the degree to which water bodies stratify. Thus, deep marshes not only should be more effective as settling basins during quiescent conditions, but should experience relatively less resuspension during wind and storm events. Finally, water in shallow marshes has greater contact with the sediments, and a shorter range for diffusion of groundwater from sediment to surface waters. Thus, the finding that shallow marshes had higher ionic strength and nutrient concentrations than wetland ponds could be predicted based on morphology alone.

Effects of retention time on wetland water quality were also detected in comparisons of wetlands with differing hydrologic type, i.e. wetlands in isolated basins had significantly higher nutrient, dissolved organic carbon and color than those with intermittent or continuous flow. Brown (1985a,b) observed that trapping efficiencies for particulate and particle-associated N and P varied as a function of water residence time both within a single wetland over time and among different wetlands. Detenbeck *et al.* (1993) found that lakes downstream from seasonally flooded herbaceous wetlands (riparian zones) or wet meadows were relatively more eutrophic and highly colored than those downstream from cattail marshes, which have a longer retention time, and thus a greater trapping efficiency.

Differences observed in Wright County wetland nitrogen speciation between 1989 and 1990 are similar to changes observed following reflooding of drained wetlands, although changes in groundwater vs surface waters differed from patterns observed by others (Cook and Powers, 1958; Kadlec, 1986). Wright County wetlands were exposed to drought conditions during the growing seasons of 1988 and 1989, followed by approximately 30% greater precipitation during 1990. The decrease in total dissolved nitrogen of groundwater samples combined with a significant increase in surface water nitrate + nitrite and particulate N between spring 1989 and spring 1990 suggests that mineralization of dissolved nitrogen increased as the water table rose, and that the ammonia formed was subsequently volatilized or transformed through nitrification or plant uptake. Duckweed blooms were observed in shallow and deep marshes in 1990, and these could account for the significant increase in particulate nitrogen. Duckweed has a relatively high N uptake rate, due in part to the high surface area/biomass ratio (Whitehead *et al.*, 1987). In contrast to our study, Kadlec observed no significant changes in dissolved nutrient concentrations of marsh cell surface waters following flooding, in spite of a significant increase in groundwater ammonium, total dissolved nitrogen, soluble reactive phosphate, or total dissolved phosphorus following macrophyte death and decay. However, filamentous algal blooms developed in flooded cells, so that nutrient uptake may have masked increased nutrient concentrations in surface waters. Likewise, Cook and Powers (1958) measured a consistent increase in ammonium in sediments of three marshes in upstate New York following flooding. Vegetative dieback in these systems was followed by blooms of duckweed or filamentous algae. Similarly, Schoenberg and Oliver (1988) observed blooms of filamentous algae as water depths increased following a dry period.

5.3. Spatial Variability of Water Quality within Wetlands

For the two wetlands sampled intensively, only specific conductivity showed significant edge effects, i.e. a significant relationship with distance from the edge independent of depth effects. Within the wetlands sampled along transects, only specifc conductivity showed a consistent trend along the transect. Specific conductivity typically shows a strong gradient both at wetland/upland and wetland/open water ecotones (Holland *et al.*, 1990), and may be related to zones of groundwater discharge (Carter and Novitzki, 1988; Winter, 1989).

Within wetland 19–3–2, water depth exhibited small-scale patchiness, and dissolved oxygen, soluble reactive phosphate, and total dissolved N followed this trend. The linear decrease in dissolved organic carbon and both inorganic and organic forms of N as a function of water depth was probably the result of lesser contact between surface water and sediments (longer diffusion pathway), greater photo-oxidation in open water, and the greater influence of algal uptake in open water. The relatively greater fraction of dissolved inorganic N in the form of nitrate + nitrite as water depth increased probably reflected an increase in the rate of nitrification relative to dentrification in the oxygenated surface waters.

Within wetland 6–3–3, dissolved oxygen appeared to be the most important driving variable as opposed to water depth or distance from the edge. Only dissolved organic C decreased as a function of water depth, presumably because of lower exchange between surface water and sediments. Color decreased not as a function of water depth, but as a function of dissolved oxygen depletion, possibly because the rate of litter decomposition was limited (Brinson *et al.*, 1981). In this wetland, the fraction of total dissolved nitrogen in inorganic form decreased as dissolved oxygen increased up to 4 mg/l, but this decrease was not mirrored by a build-up of dissolved organic N. The decrease in the fraction of mineralized N probably is related to the interaction of algal productivity with dissolved oxygen and pH. In areas where algal productivity is high, dissolved oxygen, algal uptake of ammonium or nitrate, and pH will increase. Significant amounts of ammonia can be lost through volatilization at high pH, where ammonia is in un-ionized form (Kadlec, 1979; Bowden, 1987). This phenomenon can be especially important for wetlands in agricultural zones, where loadings of ammonium from surrounding fields are extremely high. The small-scale patchiness of ammonnium was probably a sampling artifact, as extremely high values occurred when surface water was so shallow that surficial peaty sediments had to be compressed to obtain a sufficient sample volume. Ammonium values for these sites probably reflect extremely high ammonium concentrations in the sediment porewater. The proportion of dissolved inorganic N present as nitrate +

nitrite increased as a function of dissolved oxygen because denitrification required anoxic conditions.

Spatial autocorrelation affected our ability to detect meaningful patterns through CCA. The results of CCA for wetland 6–3–3 reflected spatial gradients resulting from differences in surrounding land-use rather than consistent effects of driving variables. CCA should be used with care when samples are collected at distances over which variables exhibit spatial autocorrelation. In this case, results of multiple linear regression analysis were more useful in detecting meaningful cause-and-effect relationships. Simple linear regressions can be corrected for spatial autocorrelation through procedures analogous to partial correlation analysis, i.e. by factoring out the effect of sample distance (Mantel, 1967; Legendre and Troussellier, 1988), but this correction technique has not yet been refined for use when there are two or more independent variables.

5.4. Implications for Monitoring and Experimental Designs

Our results suggest that development of monitoring and experimental designs for wetlands include the following elements.

5.4.1. *Paired Before-and-After Comparison Designs*

Our results suggest that paired before-and-after comparisons designs are likely to be more powerful overall in detecting effects of perturbations on water quality of marshes than are completely randomized designs.

5.4.2. *Use of Land-use, Wetland Class, and Hydrologic Class as Factors for Blocking, Pairing or Covariate Analysis*

Selection of a subpopulation of wetlands for sampling must take into account the following sources of variability: (1) land-use (agricultural, urban/residential, undeveloped), (2) wetland class, and (3) hydrologic type (degree of hydrologic isolation). In addition, comparison of surface water quality with groundwater quality can produce biased results unless appropriate controls are used. More powerful monitoring or experimental designs could be constructed by blocking by one or more of these factors, comparing paired reference and treatment sites based on similarity in surrounding land-use, or using watershed land-use variables as covariates in analyses.

5.4.3. *Consideration of Relative Degree of Variability Among Different Water Quality Parameters*

If sampling is to be conducted over more than one season and/or year, the potential change in wetland water quality over time must be taken into account. The concentrations and dominant form of nitrogen (ammonium, nitrate + nitrite, dissolved organic N, particulate N) are particularly sensitive to climatic changes.

The power of detecting effects in a completely randomized design was not predictable from one year to the next. The relative power of tests depends on the coefficient of variation for the different water quality measurements, which depends in turn on those driving factors influencing each measurement. For example, the power of tests to detect effects on water quality variables influenced by edaphic factors (specific conductivity, alkalinity), climatic factors (temperature), or detritivory (dissolved organic carbon, total organic carbon) was much greater than of tests to detect effects on water quality variables influenced by biotic and redox factors (nitrate + nitrite, ammonium).

Within a wetland, variation in water quality differs widely among different measurements (c.v. = 6–249%). In the two wetlands sampled, soluble reactive phosphate and ammonium were highly variable and exhibited relatively small scale patchiness.

5.4.4. *Stratified Random Sampling Within Wetlands*

Variation in dissolved organic carbon and nitrogen speciation could be explained as a function of water depth and dissolved oxygen, respectively, in the shallow marsh, while in the deep marsh most water quality variations were explained as a function of water depth or distance from the edge. Thus, one sampling stategy to reduce sampling error would be to sample several sites within a wetland, stratified by depth zone, dissolved oxygen, or distance from the edge.

5.4.5. *Use of 0.5–1.0 Mean Radius of Wetland as Intersample Distance*

The scale of spatial variability differed both among parameters and between wetlands. Scales of spatial variability can be expected to vary as a function of microtopography when water depth is a critical variable. In addition, the scale of spatial autocorrelation can be expected to vary with the size of a wetland because dispersion coefficients are scale-dependent (Fetter, 1988). The average scale over which water quality parameters were correlated was within an order magnitude of the average wetland radius in the two sites examined; however, data on more wetland sites are needed before broad generalizations can be made. Intensive sampling at a finer scale will produce redundant information if the goal is only to measure mean and variance of wetland water quality. To avoid obtaining data which exhibits spatial autocorrelation and wasting effort, the distance between sampling sites should be equal to approximately 0.5–1.0 mean radius of the wetland.

5.4.6. *Compositing Samples, Except for Those Variables with Skewed Distributions*

Of all the water quality variables measured, only the estimate of ammonium differed significantly between transect composites and transect averages. Ammonium levels were highly variable, and averaging values from a highly skewed distribution will produce an estimated mean higher than the median value. If the primary interest is to obtain a mean value for each wetland, water samples can be composited to reduce analytical costs, except for those variables with log-normal distributions (e.g. ammonium) where a biased estimate might be obtained.

6. Conclusions

Paired comparisons designs are likely to be more powerful than completely randomized designs for detecting effects of experimental treatments or environmental change on wetland water quality. Selection of a subsample of wetlands for sampling must take into account the following sources of variability: wetland type, wetland hydrologic class, and surrounding land-use. The power of detecting effects for given water quality variables will change between years, but the relative power for a test can be predicted based on primary forcing factors:

$$\text{climatic factors} > \text{edaphic factors} > \text{detritivory} > \text{land}-\text{use factors} > \text{biotic}-\text{redox or other multiple factors}$$

Subsampling within a wetland should be performed in a stratified random fashion, with stratification by depth zone, dissolved oxygen, or distance from edge (conductivity). Special techniques will be required to detect spurious correlations among water quality and environmental variables within a wetland that result only from spatial autocorrelation. To avoid obtaining data which exhibit spatial autocorrelation and wasting effort, the distance between sampling sites should be equal to approximately 0.5–1 mean radius of the wetland. If the primary interest is to obtain a mean value for each wetland, water samples can be composited to reduce analytical costs, except for those variables with log-normal distributions (e.g. ammonium) where a biased estimate might be obtained.

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